An integrated assessment of two decades of air pollution policy making in Spain: Impacts, costs and improvements.

Michel Vedrenne , Rafael Borge, Julio Lumbreras , Beth Conlan , María Encarnación Rodríguez, Juan Manuel de Andrés , David de la Paz, Javier Pérez , Adolfo Narros

This paper analyses the effects of policy making for air pollution abatement in Spain between 2000 and 2020 under an integrated assessment approach with the AERIS model for number of pollutants (NO_x/NO_2, $PM_{10}/PM_{2.5}$, O_3, SO_2, NH_3 and VOC). The analysis of the effects of air pollution focused on different aspects: compliance with the European limit values of Directive 2008/50/EC for NO_2 and PM_{10} for the Spanish air quality management areas; the evaluation of impacts caused by the deposition of atmospheric sulphur and nitrogen on ecosystems; the exceedance of critical levels of NO_2 and SO_2 in forest areas; the analysis of O_3 induced crop damage for grapes, maize, potato, rice, tobacco, tomato, watermelon and wheat; health impacts caused by human exposure to O_3 and $PM_{2.5}$; and costs on society due to crop losses (O_3), disability related absence of work staff and damage to buildings and public property due to soot related soiling ($PM_{2.5}$). In general, air quality policy making has delivered improvements in air quality levels throughout Spain and has mitigated the severity of the impacts on ecosystems health and vegetation in 2020 as target year. The findings of this work constitute an appropriate diagnosis for identifying improvement potentials for further mitigation for policy makers and stakeholders in Spain.

1. Introduction

Since the adoption in 1999 of the Gothenburg Protocol to abate acidification, eutrophication and ground level ozone, the Government of Spain has been continuously designing and implementing regulations to abate general air pollution level and comply with its international obligations as a signing party Moreover, as a Member State of the European Union, Spain is legally bound by the European regulatory framework on air and environmental quality which has required complementary policy efforts across several years. In general the policy making process for air quality control in Spain has not been a simple one, too often conditioned by conflicts between national and regional governments in the distribution of competencies. As a result, a complete and sensible analysis of the effects of these policies not only on emissions or air quality levels, but on crucial variables such as human health, protection of ecosystems and associated costs has been absent for many years. In 2011, a scientific task force headed by the Centre for Integrated Assessment Modelling (CIAM) conducted a revision of the Gothenburg Protocol using IIASA's Greenhouse gas Air pollution Interactions and Synergies (GAINS) model (TSAP Report #11). The results of this study revealed improvements in the general air quality picture in the Spain (as in Europe) but required establishing new reduction commitments for 2020 (Amann et al , 2012).

In line with these findings, the general perception among stakeholders and public opinion is that the air quality in Spain has improved but that there is still sufficient clearance for further action An example of this is that in 2012, Spain was still facing difficulties in complying with the European limit values for annual nitrogen dioxide (NO_2) concentrations in 4 air quality management zones, particulate matter (PM_{10}) daily levels in 11 zones and ground level ozone (O_3) above target value in 51 zones (Orío et al., 2013). The purpose of this study is to evaluate the impacts and benefits of the general air quality policy making process in Spain between the adoption of the Gothenburg Protocol and the target year for the new reduction commitments in 2020. The ultimate goal is to quantify the global improvement on the air quality question in Spain in order to provide enough guidance on the effectiveness of the measures that were undertaken and those that are still to be enforced towards the 2020 target year.

This analysis will be conducted following an integrated assessment (IA) approach, due to the fact that it focuses on holistic modelling perspectives that are able to generate relevant information for environmental planning without the need to configure complex atmospheric dispersion models under long run times (Moussiopoulos et al., 2009; Oxley et al., 2013). Relying in IA allows quantifying the most relevant effects that air quality has on human and environmental health for both past and future scenarios. For the concrete case of this study, this analysis was made using the newly embedded modules for the quantification of air quality impacts and costs of the Atmospheric Evaluation and Research Integrated system for Spain (AERIS), combining diagnostic (past years) and prognostic (future years) approaches (Vedrenne et al., 2014a). Besides assessing compliance with the European Limit Values (LVs) for nitrogen dioxide (NO_2) and particulate matter (PM_{10}), the analysis focused on evaluating the consequences of air pollution such as damage on ecosystems and human health, as well as estimating the economic benefits of policy driven action.

2. Methodology

As already stated in the introduction, the evaluation of the air quality control policies undertaken by the different Spanish administrations in the two decades comprised between 2000 and 2020 was carried out following the integrated assessment approach of the AERIS model. This section is devoted to explaining the origin and preparation of the necessary input data for running the model as well as outlining the methodologies and criteria for the estimation of impacts and economic benefits, recently incorporated in AERIS.

2.1. Scenario definition

The policy making activity for air quality control in Spain between 2000 and 2020 was modelled as a set of 11 biannual scenarios that contemplate a series of concrete control measures resulting in emission reductions for 5 pollutants considered by AERIS: nitrogen oxides (NO_x), sulphur dioxide (SO_2), ammonia (NH_3), particulate matter (PM_{10}, $PM_{2.5}$) and volatile organic compounds (VOC). For the years comprised between 2000 and 2012, emission scenarios were derived from officially reported emission data as well as the concrete control strategies that limited these emissions. To be consistent with the conceptual formulation of AERIS, annual emission data for these years and the before mentioned pollutants were obtained from the Spanish National Emission Inventory (SNEI) disaggregated by SNAP activity code (MAGRAMA, 2014). The future character of the period between 2014 and 2020 made necessary to rely on emission projections which in the case of AERIS are to be quantified with the Spain's Emission Projection (SEP) model (Lumbreras et al., 2008). These projections were estimated for each of the pollutants mentioned above, considering only a situation defined by the current legislation (CLE) in order to analyse the effectiveness of the implemented plans, measures and policies in Spain. Additional considerations regarding the nature and potential evolution of the macroeconomic drivers of the projections, specifically those concerning the latest economic crisis in southern Europe were implemented according to what has been published in MARM (2009) and Torrero (2010). The aggregated national emissions (without SNAP sector distinction) of the studied pollutants across the studied years are presented in Table 1 and Fig. 1.

2.2. Control measures

In general, strategic management of atmospheric pollution through policy should be translated into a specific set of control measures that have to be realistic and applicable for a concrete target activity (Vlachokostas et al., 2011). To this respect, a number of actions and measures were identified from the different national air quality management plans that have been implemented in Spain in the past years and that are deemed responsible for the experienced trend in the national air quality situation. For the future years, it was considered that CLE prevails and that no extraordinary actions to tackle air pollution will take place. For the purposes of this study, the CLE situation in Spain is the one defined by Directive 2008/50/EC and its respective transposition to the national regulatory framework through the Royal Decree 102/2011 and Law 34/2007 (Orío et al., 2013). The concrete measures that were identified are shown in Table 2, and classified as technical (TM) and non technical measures (NTM) following the end of pipe criterion established in the GAINS modelling framework (Schucht, 2005). Due to the difficulty associated with the allocation of a reduction percentage to NTM, these were not considered for the projection of emission scenarios (D'Elia et al., 2009). The selected TM were differentiated by the administrative level that applied them (national, regional and local) according to the information submitted by Spain to the European Commission (Questionnaire 461 on Directive 1999/30/EC).

2.3. Atmospheric dispersion

In order to estimate the effects of air pollution, it is necessary to quantify the concentration levels resulting from atmospheric dispersion and chemical processes across a given domain. To this respect, the IA approach followed by AERIS allows quantifying final pollutant concentration (NO_2, SO_2, NH_3, PM_{10} and $PM_{2.5}$) through a series of transfer matrices for individual SNAP sectors as a function of a baseline scenario of emissions (in this case, the emissions reported by the 2007 version of SNEI). These transfer matrices constitute a parameterisation of the WRF CMAQ air quality modelling system configured for Spain as described in Borge et al. (2014) and allow retrieving concentration values for 4500 16 km cells deployed in a 75×60 grid centred in 40°N and 3°W which covers the entire Iberian Peninsula and parts of the neighbouring countries (Vedrenne et al., 2014a). The effects of tropospheric ozone (O_3) or the secondary fraction of particles are accounted for in AERIS through a secondary pollutant module that relates them

Table 1
Aggregated annual emissions of the studied pollutants in Spain across the studied years — (annual metric tons per year).

Pollutant	2000	2002	2004	2006	2008	2010	2012	2014	2016	2018	2020
NO_x	1,415,399	1,424,453	1,456,473	1,415,782	1,213,378	1,022,917	1,043,164	1,063,994	1,039,300	1,048,280	1,063,324
SO_2	1,616,057	1,591,735	1,377,336	1,219,739	565,916	489,434	470,181	468,127	466,069	480,273	492,815
NH_3	405,271	398,997	404,289	404,031	371,952	395,139	372,996	379,879	387,211	395,019	403,330
$PM_{2.5}$	97,764	97,109	96,074	91,993	84,745	77,173	72,302	69,521	68,655	67,884	67,238
VOC	2,413,546	2,240,359	2,286,084	2,225,590	2,014,611	1,923,981	1,090,067	1,102,374	1,119,143	1,139,234	1,164,867

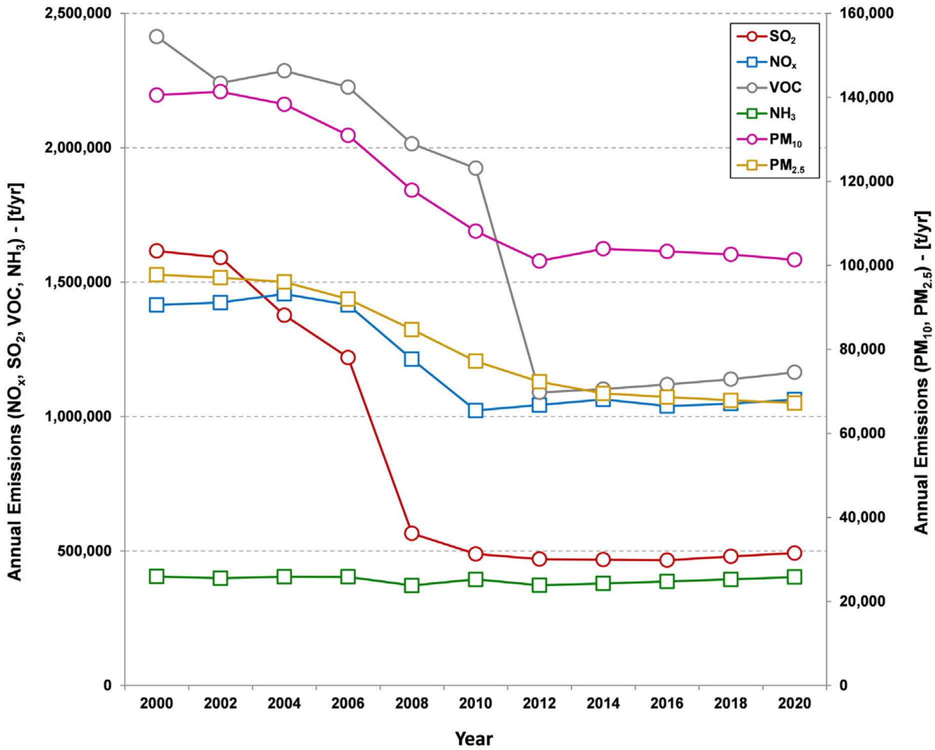

Fig. 1. Aggregated annual emission trends in Spain across studied years (2000–2020).

to the final concentrations of their respective precursors (NO_x and VOC for O_3 and NO_x, SO_2 and NH_3 for $PM_{10}/PM_{2.5}$). Concentration values can be obtained in the form of mean annual concentrations or the respective metrics that indicate compliance with the limit values (LVs) contemplated in Directive 2008/50/EC. The concentrations estimated with AERIS are comparable to those produced by its parent air quality model (WRF CMAQ), showing a good one to one statistical correspondence for all the considered pollutants (correlation coefficient $r \leq 0.97$) and a variation in predictions within 5% and 20% for annual mean concentrations (Vedrenne et al., 2014a). In order to capture the interannual meteorological variations, the transfer matrices of AERIS have been built with runs of WRF CMAQ referred to 2007 and 2014. Due to the intensive use of computational resources that is required for building a complete set of transfer matrices, the interannual variations for other years are currently reproduced in AERIS through a statistical calibration with observations from 331 background (urban, suburban and rural)

Table 2
Air quality control measures adopted by Spanish administrations between 2000 and 2012.

Sector	Measure	Type	AERIS	Admn. level
Road transport	Creation of protected atmosphere urban zones	NTM	070000	L
	Improving and promoting public transport	NTM	070100	N,R,L
	Limitation of maximum and variable circulation speed	NTM	070000	L
	Modification of the vehicle inspection procedures	NTM	070000	N
	Economic incentives for new and efficient vehicles	NTM	070000	N,R
	Renovate public fleets with efficient vehicles	NTM	070000	L
	Foster technological improvements in heavy duty vehicles	TM	0703, 0707*	N
	Incentivise the use of bicycles — bicycle paths	NTM	070100	R,L
	Increase road cleaning/washing activities	TM	070000	L
	Mobility and public transport plans	NTM	070100	N,R,L
Industry	Implementation of best available techniques (BAT)	TM	03*, 04*	N,R
	Reviewing and updating industry emission limits	NTM	03*, 04*	N
	Promoting change to cleaner fuels — end of coal funding	NTM	03*, 04*	N,R
Domestic	Domestic installation retrofitting	TM	020202	R,L
	Promoting change to cleaner fuels — discourage solid fuels	NTM	020202	N,R,L
	Change from individual to district heating	TM	020202	R,L
Energy	Implementation of best available techniques (BAT)	TM	010000	N,R
	Promoting change to cleaner fuels — end of coal funding	TM	010000	N,R
	Economic incentives to renewable energy sources	TM	010000	N,R
Construction	Limiting dust emissions through technology improvements	TM	080800	N,R,L
	Implementation of best available techniques (BAT)	TM	080800	N,R
Airports	Fleet renewals	NTM	080500	N
	Operational measures (approach, towing, boarding)	TM	080500	N
Agriculture	Banning agricultural waste burning	NTM	100500	N,R,L
	Reduce emissions associated with fertilisers	TM	1001, 1002	N,R
	Reduce emissions from manure operations	TM	100500	N,R
	Reduce engine operation times for tractors	NTM	080600	N,R

monitoring locations across Spain and Portugal. Observations from different years were available at these monitoring stations for the period comprised between 1987 and 2012, so a wide range of meteorological conditions could be reflected when this approach is applied (Vedrenne, in press). The process for building transfer matrices from air quality simulations will be revisited in the future to account for updated information (e.g., emission inventories).

2.4. Estimation of impacts

The estimation of impacts is essential for evaluating whether policies resulted in an abatement of the damage to the population, ecosystems and property produced by air pollution. The ability to describe these impacts is one of the most important advantages of relying on IA tools for the evaluation of the two decades of policy making in Spain. The following sections are dedicated to the description of the methodological approaches incorporated into AERIS for the description of such impacts.

2.4.1. Deposition and protection of ecosystems

AERIS is capable of estimating the annual accumulated atmospheric deposition of oxidised and reduced nitrogen as well as sulphur in milligrammes per hectare annum. It consists in a parameterisation of the complex deposition mechanism described by CMAQ, which quantifies wet and dry deposition of 8 different species of oxidised nitrogen, 3 of reduced nitrogen and 5 of sulphur (Im et al., 2013). The general methodology that was followed to develop such a parameterisation is published in Ganev et al. (2008) and modified to the particular conditions of the Iberian Peninsula according to Martín et al. (2011). The comparison of the estimates of accumulated deposition of species with AERIS and CMAQ revealed a good performance, with a correlation coefficients of 0.99 for oxidised nitrogen, reduced nitrogen and sulphur and an expected variation in predictions between 7% and 8% (Vedrenne, in press).

The negative effects of atmospheric deposition on ecosystems have, namely eutrophication and acidification are quantified in AERIS through the exceedance of a critical load (CL) (Posch et al., 2001). CLs are indicators of ecosystem sensitivity with a particular relevance under the framework imposed by the Convention of Long range Transboundary Air Pollution (CLRTAP) and the revision of the National Emission Ceilings Directive (NEC) (Reinds et al., 2008) and defined as an estimate of an exposure below which significant harmful effects do not occur at present knowledge (Hettelingh et al., 2013). AERIS is able to quantify exceedances according to two types of CL: for eutrophication effects through the CL for nutrient nitrogen $CL_{nut}(N)$ and for acidification effects through an acidity trapezoidal function of CL for sulphur and nitrogen ($CL_{max}(S)$, $CL_{min}(N)$ and $CL_{max}(N)$) according to the general methodology presented in Posch et al. (2001). Additionally, the current version of AERIS is able to quantify impacts caused by deposition on ecosystems in the form of an accumulated average exceedance (AAE) in order to account for ecosystem area gap closures.

The estimation and mapping of CL in the Iberian Peninsula were derived from atmospheric deposition estimates produced by GAINS model (IIASA) and the Coordination Centre for Effects (CCE) for a given and well known emission scenario and further processed statistically. This process was carried out through the combination of maps of soils, land cover and forest growth regions (Schelhaas et al., 1999; Panagos et al., 2011; Henry and Aherne, 2014).

2.4.2. Impacts on crops and vegetation

The harmful effects of air pollution on plants and trees are also taken into account by AERIS, focusing on the damage to forests caused by the exceedance of critical levels (C_l) of NO_2 and SO_2 as well as on the decrease in crop production caused by O_3 for a number of plant species. The modelling approach for describing the impacts of O_3 on crops is based on the measure of risk described by exposure response (E R) functions (Alonso et al., 2008). To this respect, E R functions based on the daylight accumulated dose over a 40 ppb threshold (AOT_{40}) were obtained for the most important agricultural and horticultural crops that Spain produces, namely wheat (*Triticum aestivum*), maize (*Zea mays*), rice (*Oryza sativa*), potato (*Solanum tuberosum*), sunflower (*Helianthus annuum*), tomato (*Solanum lycopersicum*), grapes (*Vitis vinifera*), tobacco (*Nicotiana tabacum*) and watermelon (*Citrullus lanatus*) (Mills et al., 2007; Mirasgedis et al., 2008; Van Dingenen et al., 2009). Spatial crop productions were derived as a function of harvested areas and gross crop production (NUTS 2 statistical level), incorporating specific geographic covers from EarthStat and CORINE Land Cover 2002 according to Monfreda et al. (2008). The general approach for the estimation of crop losses based on AOT_{40} values has been previously analysed in de Andrés et al. (2012) and Vedrenne et al. (2014b) for wheat.

The evaluation of impacts on forests is made by inspecting the degree of exceedance of the critical levels (C_l) of NO_2 and SO_2, defined as a situation when either the annual mean concentration or the winter half year mean concentration for the before mentioned pollutants is greater than 30 $\mu g/m^3$ and 20 $\mu g/m^3$ respectively. These exceedances are only quantified for those cells which contain one or more of the following forest categories according to CORINE Land Cover 2002: broadleaved deciduous forests, broadleaved evergreen forests, mixed leaf type forests and needle leaved evergreen forests (Pilli, 2012).

2.4.3. Impacts on human health

In the latest version of AERIS, the estimation of the health related impacts is limited to particulate matter ($PM_{2.5}$) and tropospheric ozone (O_3). The general conceptual framework that has been followed is the one addressed by GAINS and published in Mechler et al. (2002) and Rao et al. (2012). This methodology allows AERIS to provide estimates on the change in statistical life expectancy per person (Δe_c), the total amount of life years lost (YOLL) and the disability adjusted life years (DALY) for $PM_{2.5}$ related cardiopulmonary impacts and the annual cases of premature mortality (Mort) produced by exposure to O_3. The estimation of health affectations by air pollutants depends on the quantification of the respective relative risks, whose values in this case stemmed from two epidemiological studies for exposure to $PM_{2.5}$ (Pope et al., 2002) and to O_3 (Jerrett et al., 2009; Heal et al., 2013). The estimation of Δe_c and YOLL requires the definition of a survival function, which for the case of AERIS was determined from mortality rates which were taken from the World Health Organization (WHO) mortality data and statistics database for cohorts aged between 30 and 60 in Spain, Portugal, Andorra, France, Algeria and Morocco between the years 1990 and 2010 (WHO, 2014). The quantification of cardiopulmonary[1] DALY required the definition of raw disability values obtained from the Mortality and Burden of Disease Estimates for WHO Member States in 2004 under egalitarian principles (Mathers et al., 2006). Additional considerations for the estimation of health impacts in Spain were made according to Boldo et al. (2012). Discussions on the representativeness of the estimates that AERIS provides on the before mentioned health impacts have been published in Vedrenne et al. (2014b).

2.5. Estimation of costs and benefits

The valuation of air pollution costs and benefits associated to policy making in AERIS is presently carried out only for health and non health damage variables in millions of euros per year (M€/yr). The cost to individuals and health services due to air pollution disability (restricted activity days) was modelled for $PM_{2.5}$ under an impact pathway approach adapted to Spain (Dixon, 1998; Defra, 2013). The valuation of non health damage was carried out for crop losses (O_3 related) and building soiling ($PM_{2.5}$) according to Watkiss et al. (2001). Market prices for

[1] WHO GBD codes: 39, 40, 106, 107, 108, 109, 111.

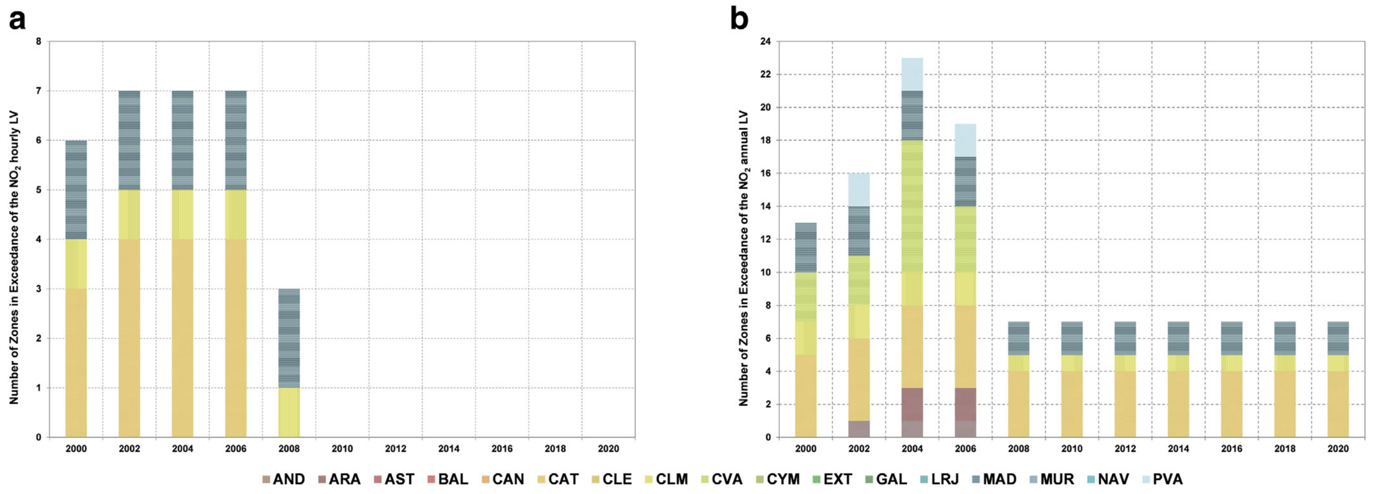

Fig. 2. Number of air quality management zones above the NO_2 a) hourly and b) annual LVs between 2000–2020 simulated with AERIS.

cereals (wheat, maize and rice) were taken from local data published by the regional government of Castille and prices for the rest of the crops (at origin) were drawn from the Ministry of Finance (JCL, 2014; MEC, 2014). Costs were estimated for the 11 biannual scenarios and referred to 2014 using a nominal annual interest rate published by the Bank of Spain of 0.55% (BDE, 2014). Prices were increased in 2.5% per annum to account for inflation, and further increased by 2% per annum to reflect the assumption that willingness to pay for health will rise in line with economic growth assuming a similar behaviour in Spain to that of the United Kingdom (Defra, 2011). A mean Sterling to Euro exchange rate was calculated from the 2012 daily spot exchange rates published by the Bank of England (BE, 2014). Damage costs on ecosystems, soils or water bodies are not being presently addressed in AERIS.

3. Results and discussion

3.1. Analysis of air quality levels and compliance with Directive 2008/50/EC LVs

As it was already introduced in Section 2.1, the different sets of mea sures implemented by the Spanish authorities at the three administra tive levels (national, regional and local) have resulted in a general decrease in the emission or airborne pollutants (NO_x, SO_2, NH_3, PM_{10}, $PM_{2.5}$, VOC) in the period between 2000 and 2014, while the period be tween 2014 and 2020 does not experience a substantial change in the emissions for these pollutants (Fig. 1). These small changes in emissions can be explained by the fact that most of the policies considered to be the 'low hanging fruits' have already been applied in Spain (e.g., limita tion of the sulphur content in fuels) (Amann et al., 2011). Moreover, it has been assumed that the deterioration of the economy in Spain due to the financial crisis will continue to be an obstacle for any additional substantial reductions in the emissions of pollutants, mostly caused by budget (public and private) limitations or unfavourable behaviour changes in the population (Saffari et al., 2013).

Modelling such emission reductions with AERIS allowed assessing compliance for the 126 air quality management zones of Spain (exclud ing the Canary Islands)[2] for the hourly and annual LVs for NO_2 and for the daily and annual LVs for PM_{10}. Despite the fact that other pollutants are regulated by Directive 2008/50/EC, analysis was limited to NO_2 and PM_{10} as only compliance with their respective LV was analysed in IIASA's TSAP Report #11. Fig. 2 shows the number of zones in exceedance of the NO_2 LVs for the period between 2000 and 2020. For both LVs, a decrease in the number of exceeding zones can be seen across the studied years, from a total of 6 zones in infringement of the NO_2 hourly LV in 2000 to no zones in 2020 and from 13 zones in in fringement of the NO_2 annual LV in 2000 to 5 zones in 2020 (with max ima being 7 and 21 zones respectively). Most of these zones are located in Catalonia (CAT) and Madrid (MAD) on which the largest urban ag glomerations in Spain are located (Madrid and Barcelona). These two communities account for 5 and 3 air zones above the NO_2 annual LV in 2020. The number of zones in infringement of the PM_{10} LVs between 2000 and 2020 are shown in Fig. 3. Although similar to the number of zones above the NO_2 LVs, the decrease in the number of zones in fringement of the PM_{10} LVs has been modest (28 zones in 2000 against 21 zones in 2020 above the daily LV and 3 zones an 2000 against no zones in 2020 above the annual LV). Most of the zones in infringement with the PM_{10} daily LV are located in Andalusia (AND), Asturias (AST), Catalonia (CAT), Galicia (GAL), Madrid (MAD) and Murcia (MUR), while the zones above the annual LV concentrate in Catalonia (CAT), the Valencian Community (CVA) and Galicia (GAL). In summary, the de crease in the air quality zones in infringement is in line with the de creasing air quality trends observed in Spanish monitoring locations for NO_2 and PM_{10} (Querol et al., 2014). Due to the inability of AERIS to quantify the natural contribution of Saharan dust to the general air qual ity levels, no discounts on the PM_{10} concentrations have been made to reduce the number of zones in non compliance. This subtraction is crit ical for those zones located in Southern Spain, which are especially vul nerable to high dust episodes (de la Paz et al., 2013).

The number of zones quantified by AERIS for Spain (Figs. 2 3) con sists in a conservative estimate that does not take into consideration the different local factors (i.e., meteorology, emplacement, pollution hotspots) that drive monitoring locations to register high concentra tions. This results in the fact that the estimate of zones in infringement by the model does not necessarily coincide with the official number of zones that exceeded the NO_2 and PM_{10} LVs between 2004 and 2012 ac cording to Questionnaire 416. For example, in 2012 the number of zones above the NO_2 hourly LV quantified by AERIS was 0, while Spain official ly reported 2 zones in infringement (ES1301 and ES1309). Despite the abovementioned, the estimates of AERIS are underpinned by a robust parameterisation of a CMAQ system applied to the Iberian Peninsula, which has been validated against observations (Vedrenne et al., 2014a,b) and which in turn has been used for assessing the compliance of the air quality management zone of the city of Madrid (ES1301) with the NO_2 LVs (Borge et al., 2014). The evolution of the rest of pollutants simulated by AERIS (SO_2, $PM_{2.5}$, NH_3, O_3), which are also regulated by Directive 2008/50/EC through limit or threshold values, will be analysed

[2] Considering a total number of zones in Spain of 134 as of 2010.

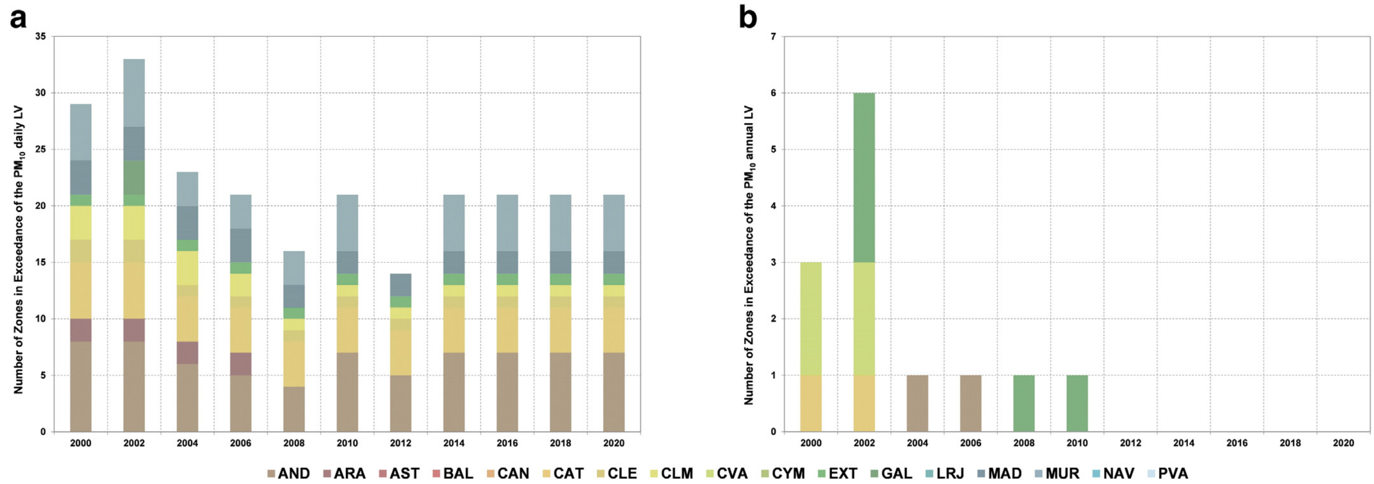

Fig. 3. Number of air quality management zones above the PM_{10} a) daily and b) annual LVs between 2000–2020 simulated with AERIS (without natural subtraction).

indirectly in the following sections through the different impacts they produce on other aspects such as health or ecosystems.

3.2. Analysis of the contribution of the autonomous communities to air quality levels

It is often found that air quality management zones do not achieve compliance because of the contribution of transboundary air pollution from the nearby regions. As a consequence, the average contributions of the individual autonomous communities to the air quality levels of NO_2 and $PM_{2.5}$ between the years 2000 and 2020 are presented (Fig. 4). As it can be seen, the degree of contribution of the different au tonomous communities is different and pollutant specific. The annual means of NO_2 of most of the autonomous communities are largely de termined by the emissions located in their territory, ranging from 62% for the Basque Country (PVA) and 99% for the Balearic Islands (BAL) (Fig. 4a). In the case of the annual means of $PM_{2.5}$, the degree of contri bution of the emissions located in the territory of the autonomous com munity is variable but not as high as for NO_2. The lowest degree of contribution is observed for Castille and Leon (CLE 12%) and the highest for Ceuta and Melilla (CYM 98%). The case of contributions to the $PM_{2.5}$ levels for CLE is extreme, as annual means are the product of the emis sions produced in 10 communities (Aragon, Asturias, Cantabria, Castilla La Mancha, Extremadura, Galicia, La Rioja, Madrid, Navarre and the Basque Country) (Fig. 4b). The information provided by the contribution analysis is able to hint on the origins of air pollution at a given air quality management zone in order to orientate efforts from local authorities towards abating emissions from sources under their control.

3.3. Analysis of deposition and impacts on ecosystems

The reduction in emissions registered in Spain between 2000 and 2020 brought about a reduction in the atmospheric deposition of oxidised nitrogen (N_{ox}) and sulphur (S), as well as a slight increase in the deposition of reduced nitrogen (N_{red}). The reduction of sulphur de position equalled 63% in the studied period, from an initial deposition of 550,000 $mg/m^2 \cdot yr$ in 2000 to approximately 200,000 $mg/m^2 \cdot yr$ in 2020. The evolution of the deposition of oxidised nitrogen was modest, descending from an initial 240,000 $mg/m^2 \cdot yr$ in 2000 to a final 210,000 $mg/m^2 \cdot yr$ (14%), while the deposition of reduced nitrogen in creased from 180,000 $mg/m^2 \cdot yr$ in 2000 to 270,000 $mg/m^2 \cdot yr$ in 2020 (+50%). While the decrease in the atmospheric deposition of S and N_{ox} is directly related to a decrease in the emissions of SO_2 and NO_x in the studied years (69% and 24% respectively Fig. 1), the increase in the deposition of N_{red} is associated with the deposition chemistry considered by CMAQ, which in limited conditions of SO_2 and NO_x (as a product of emission reductions) and almost constant levels of NH_3 (0.5% between 2000 and 2020), enhances its deposition as ammonium species (Sotiropoulou et al., 2004).

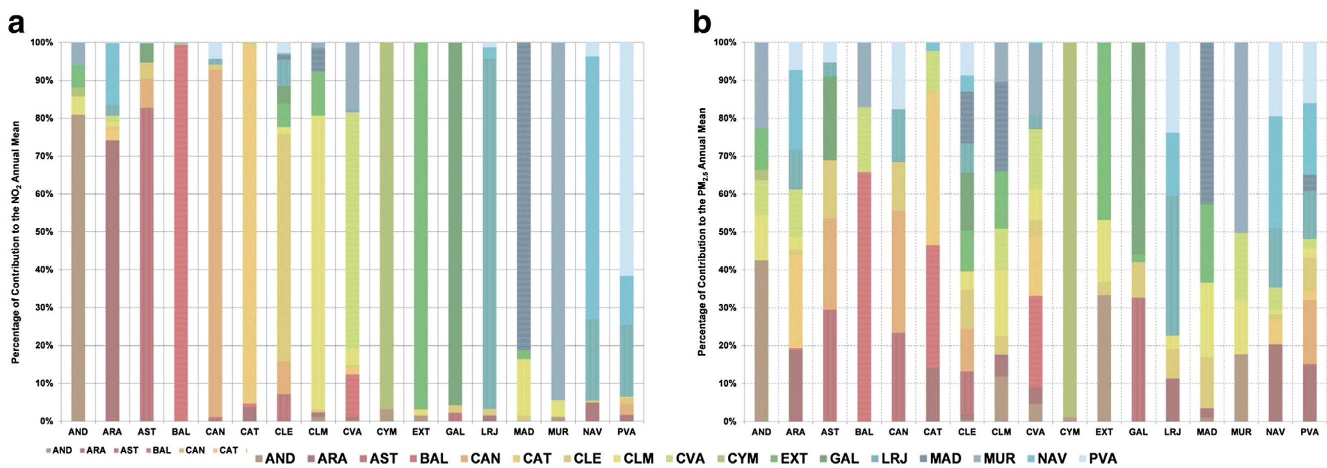

Fig. 4. Average percentage contributions of the individual autonomous communities to a) NO_2 and b) $PM_{2.5}$ annual means in 2000–2020 simulated with AERIS.

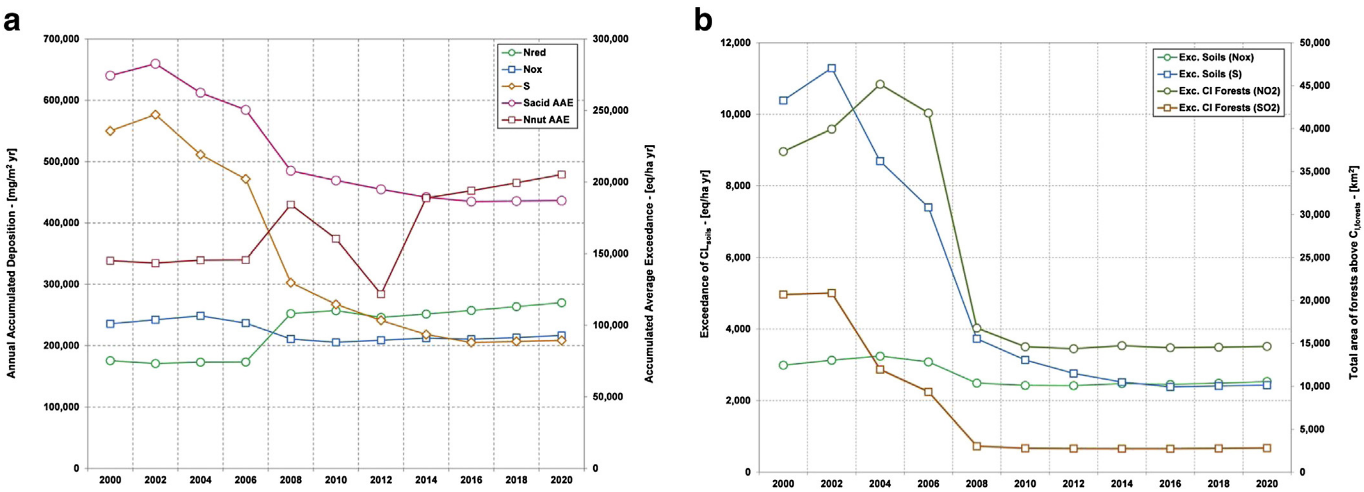

Fig. 5. Temporal evolution in 2000–2020 of the a) annual accumulated deposition per unit of surface of N_{red}, N_{ox}, S and AAE for acidification sulphur (S_{acid}) and nutrient nitrogen (N_{nut}); b) annual exceedance of critical loads for soils due to deposition of N_{ox} and S and total area of forests above C_l of NO_2 and SO_2.

The reduction in the total deposition budgets registered for the years between 2000 and 2020 produced a similar tendency in the respective excess depositions (as average accumulated exceedances). In the case of the excess acid deposition ($S_{acid,AAE}$), a clear decrease has been witnessed in Spain from an initial 640,000 eq./ha·yr in 2000 to a final 430,000 eq./ha·yr in 2020 (32%). The case of the excess nitrogen deposition (N_{nut},AAE) shows an increase in the studied year from 325,000 eq./ha·yr in 2000 to 490 eq./ha·yr in 2020 (+51%), after reaching a minimum deposition in 2012 of 280,000 eq./ha·yr (Fig. 5a). In the case of excess acid deposition of nitrogen and sulphur species on soils ($Exc_{Soils,NOx}$, $Exc_{Soils,S}$), decreases are observed between 2000 and 2020. In the case of $Exc_{Soils,NOx}$, excess deposition was reduced from 3650 eq./ha·yr to 2500 eq./ha·yr (31%) while for $Exc_{Soils,S}$ the decrease was more noticeable, namely 72% (9000 eq./ha·yr in 2000 against 2500 eq./ha·yr in 2020). In general, it was observed that policy directed towards abatement of airborne pollutants has reduced significantly the impacts of sulphur on ecosystems due to deposition, while the impacts caused by nitrogen species experienced a more modest reduction. In every case, the reduction patterns of the impacts on ecosystems seem to correspond with those of emission abatements between 2000 and 2020.

3.4. Analysis of impacts on crops and vegetation

The abatement of NO_x and SO_2 emissions in the 20 year period between 2000 and 2020 has reduced the phytotoxic impact that these pollutants have on forests. The decreasing trend of forest area above NO_2 and SO_2 critical levels (C_l) is shown in Fig. 5b. In 2000, forest land above 20 μg/m^3 of SO_2 was approximately 22,000 km^2 and reached a minimum level in 2008 of 2500 km^2 (88%) which will be kept unaltered until 2020. The reason for this substantial decrease in the exposition of forests to SO_2 is the ban imposed on high sulphur gasoil in 2000 and its further revisions in 2005 and 2009 which reduced the emissions of sulphur into the atmosphere. The area of forests above 30 μg/m^3 of NO_2 showed a moderate change in the period between 2000 and 2020, from approximately 13,000 to 10,000 km^2 (23%). Both tendencies seem to follow an analogous evolution pattern with that of SO_2 and NO_x emissions (Fig. 1).

Reductions in the emissions of NO_x and VOC have reduced crop damage due to vegetal exposure to O_3. Fig. 6 depicts the trend of these reduced impacts in terms of the monetary losses due to crop damage in Spain. It is worth noting that the comparison was carried out in economic terms due to the different annual yields of the individual crops to attenuate the effect of the cultivated land extension and to reflect their market value. In 2000 the total losses due to O_3 induced crop damage accounted 20.7 million €$_{2014}$, which were reduced to 18.8 million €$_{2014}$ in 2020 (11%). The bulk of economic losses concentrate on wheat (10.5 m€$_{2014}$ in 2000 vs. 9.6 m€$_{2014}$ in 2020), followed by potato (2.9 m€$_{2014}$ in 2000 vs. 2.7 m€$_{2014}$ in 2020) and grapes (2.6 m€$_{2014}$ in 2000 vs. 2.3 m€$_{2014}$ in 2020). The total mass of wheat lost in the studied period accounted 510,000 metric tonnes, with 35% of these losses concentrated in Castilla y León only (180,000 tonnes). Potato losses added up to a total of 120,000 tonnes, concentrating mainly in Galicia (18%) while total grape losses were 118,000 tonnes with 45% of them occurring in Castilla La Mancha (54,000 tonnes). Policy making activity that has taken place in the last 20 years in Spain has prevented farmers and producers from losing 1.9 m€$_{2014}$, which could have been externalised to consumers through increases in food prices. Additionally, the abatement of emissions has increased commodity offer in approximately 2 million metric tonnes.

3.5. Analysis of impacts on human health

The quantification of impacts on human health is an essential component for air pollution integrated assessment. Stakeholders might be interested in assessing the effectiveness of policies directed towards emission abatement and control on public health, which is also paramount in decision making (Tasić et al., 2012). The emission reductions between 2000 and 2020 have produced benefits in human health for the two pollutants characterised by epidemiological studies: $PM_{2.5}$ and O_3 (Fig. 7). The impacts of O_3 in mortality (expressed as the number of deaths) in Spain have not improved in the studied period; the modelled mortality in 2000 (732 cases) was marginally increased in 7 additional deaths in 2020. The highest improvement is observed in the period between 2000 and 2020 (6 less mortality cases), while the worst period was between 2006 and 2008 (4 additional deaths to the 2000 levels and 14 deaths compared to the previous biannual period). The highest mortality cases are allocated in the autonomous communities of Andalusia (AND), Catalonia (CAT) and Madrid (MAD). This is related to the presence of large urban centres (i.e., Madrid, Barcelona and Seville) in such communities, their location in high ozone regions in Spain and their high populations (Baldasano et al., 2011; Querol et al., 2014). Additionally, the variations in the emissions of NO_x and VOC do not seem to decrease in an analogous pattern, showing a more restrictive control for VOC than for NO_x (Fig. 1). Despite this, VOC emissions are always greater than NO_x which could derive in high VOC/NO_x ratios that complicate O_3 control (Vivanco, 2009; Dimitriou and Kassomenos, 2015). The relatively negligible increase in O_3 mortality cases is also related to the presence of high regional hemispheric backgrounds in the Mediterranean basin (Heal et al., 2013).

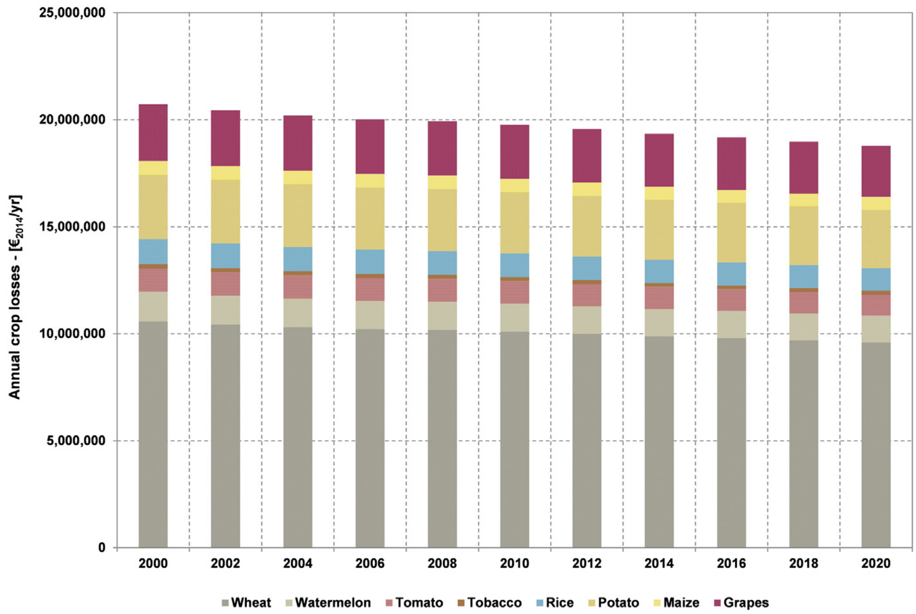

Fig. 6. Temporal evolution in 2000–2020 of the annual crop losses of grapes, maize, potato, rice, tobacco, tomato, watermelon and wheat due to exposure to O_3 (losses referred to Spanish € of 2014).

Regarding the evolution of $PM_{2.5}$ related impacts, improvements have been witnessed as a consequence of the abatement of primary particle emissions as well as NO_x, SO_2 and NH_3 which contribute to the total particle mass (Fig. 1). From Fig. 7, only one biannual period produced worse levels of mortality (measured as YOLL) than 2000 (230,700 years), namely 2000 2002 (233,200 years). After 2002 improvements with respect to 2000 have been witnessed, reaching a maximum improvement in the period between 2006 and 2008 (203,000 years and 5087 years won with respect to 2004 2006). After 2012, the number of years lost due to $PM_{2.5}$ air pollution reaches a level of approximately 195,500 years which is maintained until 2020 and registering marginal biannual improvements. The total number of years won due to air pollution management in Spain in the studied period was 36,271 years, most of which are concentrated in the autonomous communities of Andalusia (AND, 16%), Catalonia (CAT, 17%), Valencian Community (CVA, 9%), and Madrid (MAD, 18%). According to Boldo et al. (2014), $PM_{2.5}$ related mortality in Spain is heavily influenced by road traffic emissions and particularly intense in urban environments. In this sense, health improvements become more evident for these autonomous communities due to the fact that these contain the largest urban centres in Spain, and concentrate 47% of the total Spanish population. The contribution of Saharan dust to adverse health was not studied because it has been observed that particles originating from long range transport are less toxic than those originated by local sources (e.g., road traffic) (Samoli et al., 2011).

3.6. Analysis of air quality costs

Policy makers are often confronted with the need of quantifying the cost that poor air quality has on society, particularly if this has an impact on public spending. For the purposes of this paper, the costs of addressing the consequences that air quality has on crops, human health and soiling have been quantified for each of the 11 biannual scenarios (2000 2020). Savings were quantified as the differences between the incurred costs for every scenario and the costs of a do nothing scenario in 2000 (Fig. 8). These savings can be interpreted as the amount of money whose expenditure is avoided to stakeholders (either authorities or individuals) due to air quality policy making. As it can be seen from Fig. 8, the total savings accrued in the studied period due to

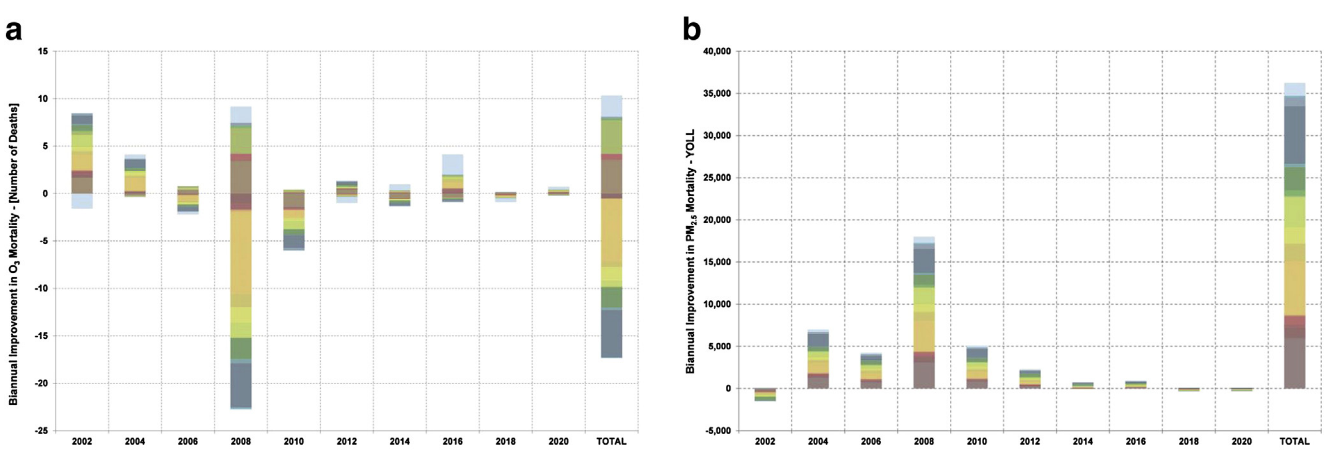

Fig. 7. Biannual and total improvements in a) O_3 mortality (number of deaths) and $PM_{2.5}$ mortality (YOLL) between 2002 and 2020 by autonomous community. Negative values denote increases in mortality.

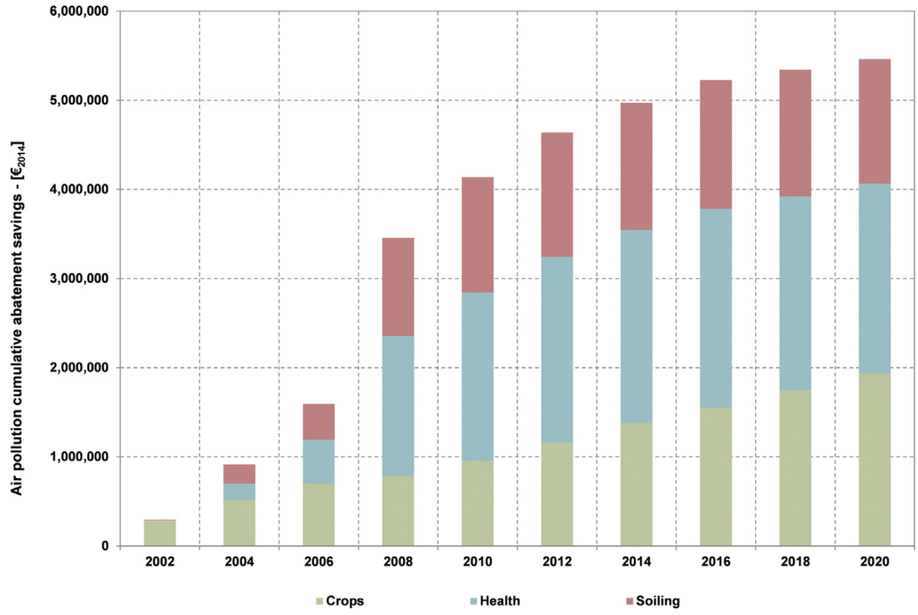

Fig. 8. Cumulative abatement savings for impacts of air pollution on crops, human health and soiling (savings referred to Spanish € of 2014).

emissions abatement rose to 5,463,362 $€_{2014}$, growing biannually since 2008 at an almost constant 13%. Of these savings, an average of approximately 40% of these savings is due to health improvements (2.1 m $€_{2014}$). The highest improvement in savings is achieved in the period between 2006 and 2008, which is in line with the improvement in YOLL for that same period as shown in Fig. 7. It should be noted that the quantification of health related costs is partial in the sense that it only considers low severity disability (restricted activity dates), whose costing is only related with decreases in wage rates due to work absence. Costs due to high severity disabilities such as hospital admissions, emergency room visits or illness in children due to $PM_{2.5}$ or other pollutants are not being considered at present.

As it was already discussed in Section 3.4, the impact that O_3 had on crops was mitigated in monetary terms produced savings of 1.9 m$€_{2014}$ in the studied period, after continuous reductions (Fig. 6). In average, the contribution of crop impact mitigation is approximately 39%. Soiling costs due to the soot fractions of $PM_{2.5}$ are associated with building cleaning and restoration and account for 25% of the total savings (1.3 m$€_{2014}$). As with health impacts the greatest improvement is observed between 2006 and 2008 (+18%). Despite the abovementioned, the savings quantified in this section are gross as AERIS does not quantify abatement costs (i.e., the costs associated with the implementation of mitigation measures).

4. Conclusions

The study of twenty years of policy making for air pollution control and management in Spain with the AERIS model for integrated assessment has allowed quantifying the effectiveness of the efforts directed towards the abatement of pollutants between 2000 and 2020. In general, the witnessed reductions in the emissions of air pollutants have brought about benefits in the different aspects that are encompassed in the air pollution question. Regarding the compliance of Spain with the European air quality limit values for NO_2 and PM_{10} specified by Directive 2008/50/EC, the number of zones in infringement exhibited a sustained decrease in the studied period. The decreases in the NO_2 levels are due principally to measures that concentrate on road traffic and general combustion processes (industrial and domestic), while the mitigation of PM_{10} concentrations is also associated with power generation and off road sources. Despite this, it has been anticipated that a number of air quality management zones will still be in infringement of such limit values under CLE scenarios in 2020 for both pollutants. The analysis of the contribution of the autonomous communities to such exceedances suggested that the air quality levels are not substantially influenced by transboundary dispersion from neighbouring communities, being these contributions generally higher for NO_2 than for $PM_{2.5}$.

Decreases in the impacts that air quality has on other aspects such as atmospheric deposition, damage on crops and forests and affectation of human health have also been witnessed. Regarding impacts on ecosystems due to atmospheric deposition (exceedance of critical loads) the affectation due to sulphur has been widely mitigated in the studied period, an issue that is directly related to the restrictions imposed on sulphur contents of fuels and to the shift to cleaner fuels in the power generation and industrial sectors. The abatement of nitrogen related deposition was more modest due to the increasing difficulty associated with controlling NO_x emissions. Analogous decreases were observed for the area of forests above critical levels of NO_2 and SO_2. The analysis of impacts on crops also evidenced a decrease in the monetary losses due to O_3 induced crop damage due to the control of NO_x and VOC emissions, avoiding the potential externalisation of approximately 1.9 m$€_{2014}$ to the commodities market. When considering health impacts, the degree of impacts produced by human exposure to O_3 revealed a very marginal increase in the 20 year period (6 additional deaths), basically due to unharmonised control strategies for VOC and NO_x that did not limit the formation of O_3 and the high hemispheric background levels in the region. Regarding $PM_{2.5}$ related health impacts, these were substantially decreased as evidenced by the number of years of life won by population groups and to the increasing savings on work related absences due to air pollution related disability. Finally, undertaking air quality actions directed towards emission abatement brought about gross savings in the order of 5.4 m$€_{2014}$.

The evaluation also revealed that impacts are still being produced and that there is still enough clearance for air quality improvements in 2020. However, it should be noted that the improvements in the general air quality of Spain between 2000 and 2020 were achieved under moderate ambition levels and without considering departures from current legislation. The results presented in this paper provide a useful starting point for identifying key issues on which further abatement can be allocated in order to deliver better air quality in Spain in the forthcoming years.

Disclaimer

The research and findings presented in this paper are responsibility of the authors and do not necessarily reflect the position or views of Ricardo AEA. The AERIS model is intellectual property of the Technical University of Madrid. There is no conflict of interest.

Appendix A

Table A.1
Sectors and pollutants currently considered by AERIS.

SNAP code	Activity name	NO_x	SO_2	PM_{10}	$PM_{2.5}$	NH_3
010000	Coal-fired power plants > 300 MW	●	●			
020202	Residential plants < 50 MW	●	●	●	●	
030000	Combustion in manufact.—area sources	●	●			
040000	Production processes—area sources		●			
070101	Passenger cars—highway driving	●		●	●	
070103	Passenger cars—urban driving	●		●	●	
070201	LDV < 3.5—highway driving	●		●	●	
070203	LDV < 3.5—urban driving	●		●	●	
070301	HDV > 3.5 t—highway driving	●		●	●	
070303	HDV > 3.5 t—urban driving	●		●	●	
0707 + 08	Break, tire and road abrasion processes			●	●	
080500	Airports LTO/yr > 10.000	●				
080600	Agriculture (machinery)	●	●	●	●	
080800	Industry (machinery)	●	●	●	●	
100101	Culture with fertilisers—permanent crops					●
100102	Culture with fertilisers—arable crops					●
100500	Other agricultural activities			●	●	●
110000	Other sources and sinks					●
–	Portugal (total)	●	●	●	●	●
–	International ship transit	●	●			

Table A.2
Spanish autonomous communities considered by AERIS.

Code	Autonomous community	Official name
AND	Andalusia	Andalucía
ARA	Aragon	Aragón
AST	Asturias	Principado de Asturias
BAL	Balearic Islands	Islas Baleares/Illes Balears
CAN	Cantabria	Cantabria
CAT	Catalonia	Cataluña/Catalunya
CLE	Castille and Leon	Castilla y León
CLM	Castille-La Mancha	Castilla-La Mancha
CVA	Valencian Community	Comunidad Valenciana/Comunitat Valenciana
CYM	Ceuta and Melilla	Ciudades Autónomas de Ceuta y Melilla
EXT	Extremadura	Extremadura
GAL	Galicia	Galicia
LRJ	La Rioja	La Rioja
MAD	Community of Madrid	Comunidad de Madrid
MUR	Region of Murcia	Región de Murcia
NAV	Navarre	Comunidad Foral de Navarra/Nafarroako Foru Komunitatea
PVA	Basque Country	País Vasco/Euskal Herria

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